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A study on contamination of ground and surface water bodies by leachate leakage from a landfill in Bangalore, India

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Abstract

This paper discusses the effects of a potential leachate leakage from a municipal solid waste landfill, situated at Mavallipura, Bangalore, India, on the surrounding water bodies. The landfill area is spread over an area of about 100 acres that began accepting waste from 2005. MSW was deposited in non-engineered manner that has resulting in steep and unstable slopes, leachate accumulation within the MSW mass, and leachate runoff into nearby water bodies such as ponds and open wells. The current study investigates the physicochemical characterization of landfill leachate and nearby water bodies. The batch leach tests were conducted to know the heavy metal concentrations in the contaminated soil. A series of column tests were also conducted to estimate the migration rates of different contaminants through the soil. Furthermore, these transport parameters were considered as input for fluidyn-POLLUSOL model to estimate the migration of leachate from the landfill site to the surrounding water bodies.

Keywords: Concentration, Landfill, Leachate contaminant

Introduction

Generally, the municipal solid waste landfills create lots of environmental pollution due to landfill gas combustion, leakage of leachate and foul smells. Among all these, leakage of leachate affects the surrounding environment the most, especially the surface and ground water bodies because the leachate consists of high concentrations of heavy metals, organic compounds and toxic contents. Recently, several cases have been reported around the world related to pollution of water bodies which were caused by municipal solid waste landfills [1].

The production and usage of heavy metals such as copper, cadmium and zinc have increased substantially over the years [2]. The excess quantity of heavy metals disposed off the land can cause significant damage to the environment and human health as a result of their mobility, solubility and their ability to transfer in water or plants [2, 3].

The leachate from MSW landfills may leak into groundwater aquifers due to rainfalls, spread into the adjacent river system by groundwater flow and pollute the surrounding environment. However, this process does not stop even after the landfill activities have stopped receiving solid waste. Hence, it is very essential to keep assessing and monitoring the surroundings of decommissioned landfill sites.

The contaminant transport must be modelled in soil and water media to estimate the level of contamination that has happened/going to be happening. Furthermore, these models are helpful in selecting a proper remediation method for the affected site.

The contaminant transport parameters can be determined through laboratory column experiments to get the accurate values or they can be estimated from the related data documented in the literature by using some correlations. The contaminant transport parameter values for different metal ions are available in the previous studies [4–17].

Currently, various specialized computer software packages have been evolved and used to address the contaminant transport problems in groundwater system. However, Fluidyn groundwater model-POLLUSOL is the most powerful software package which is based on the finite volume modelling hydraulic and concentration distribution. The other software tools like post processing particle tracking model (MODPATH), the finite difference groundwater model (FEMWATER), two-dimensional finite element model (SPEED 2D), multi-phase transport model (UTCHEM), transport tools and the transition probability geostatic software (T-PROGS) are also in use to model the contamination transport in groundwater and to predict the interaction between water bodies like surface and groundwater. The fluidyn POLLUSOL software tool is used specifically for simulating groundwater flows and predicting pollutant dispersion from a source through a porous medium. Flow equations are derived using conservation of total fluid mass principle and Darcy's law for flow through porous media. Solute transport equation is derived from conservation of mass for a single solute or multiple solute species that may decay or adsorb in the porous medium.

Soil and water contamination are the most common problem encountered at the landfill sites in India. The objective of this study was to assess the possibility of reclamation of a closed landfill site located at Mavallipura near Bangalore, India. The existing liner system at this site has not completely prevented the leachate migration to the underlying aquifers, which is required to be managed and controlled to avoid adverse impact on the surrounding environment.

The soil profile, physicochemical parameters, hydraulic head and concentration levels of different contaminants were studied at this site. The batch leaching tests were conducted to know the heavy metal concentrations in the contaminated soil. Column tests were also conducted to estimate the migration rates of different contaminants through the soil. These migration rates were determined by the analytical solution of advection–dispersion equation using MATLAB software tool. The transport rates obtained from column tests along with the field data collected at the landfill site were given as input to the fluidyn-POLLUSOL model to simulate and assess the leachate migration from the landfill. This model may be useful to analyze the leachate migration from the landfill and its associated environmental impacts, particularly on groundwater wells and surface water ponds down gradient of the site. This research study is also useful to characterize landfill leachate through physicochemical analysis. Finally, this study also aims to serve as a guideline for the implementation of an appropriate treatments/remedial measures for reducing the adverse effects on the environment.

Materials and methods

Study area

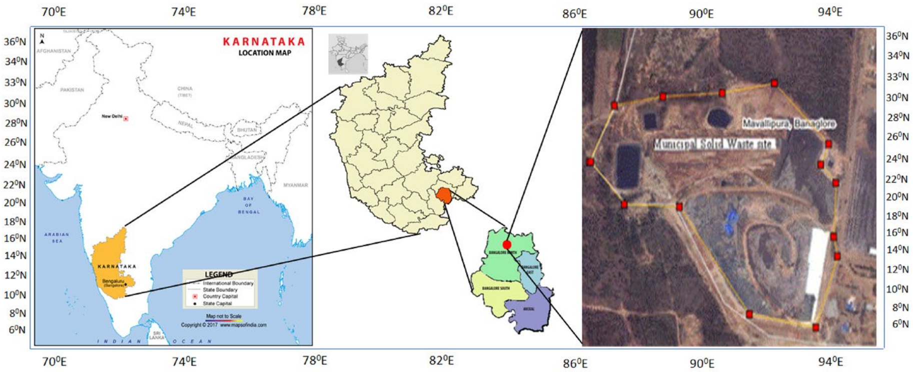
The Mavallipura landfill site is in the north of Bangalore, India at Latitude 13°50′ North, Longitude 77°36′ East in the state of Karnataka. This landfill site has been used as a processing site for the municipal solid waste generated from the Bangalore city. Bangalore receives an average annual rainfall of 978 mm on long term basis. The primary rainy season is from June to September and the secondary rainy season is from November to December. The Mavallipura village is located about 20 km away from Bangalore. About 100 acres of land in and around the village is used for dumping Bangalore's municipal waste by the Bruhat Bangalore Mahanagara Palike (BBMP–Greater Bangalore Municipal Corporation) that began accepting waste from 2005. The landfill was operated by M/s. Ramky Environmental Engineers from 2007 onwards and can sustain about 600 tonnes of waste per day. However, the BBMP has been sending almost 1000 tonnes of garbage from Bangalore city every day. Citizens around Mavallipura village have demanded to stop the landfill activities as it is illegal and unscientifically managed, and thus it is now closed for land filling. A soil liner of 0.3 m thickness has been applied historically, and the MSW is deposited in an unscientific manner that has resulted in steep, unstable slopes, leachate accumulation within the MSW mass, and leachate runoff into nearby water bodies such as ponds and opened wells.

The landfill site is about 100 acres out of which approximately 35 acres of land was used for landfill. The landfill has 3 cells at present. Cell 1 is filled and covered by a HDPE membrane. Cell 1 is spread over around 5 acres. Cells 2 and 3 are operational at present (Fig. 1). A single liner system was used in the landfill. The leachate collection systems are located outside the landfill. The MSW was dumped in the landfill in layers of 1–3 m. Waste pickers had access to recyclable materials before solid waste arrived at the landfill site. There are leachate collection sumps for the collection of the leachate. High Density Polyethylene (HDPE) pipes were used for the collection of the leachate. It is laid at a slope of 1:2. These pipes were connected to the leachate collection sumps.

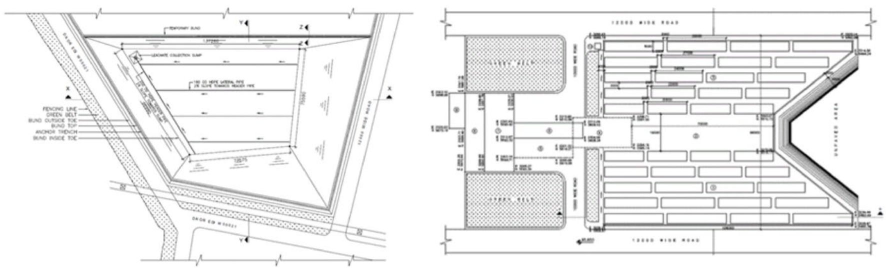
The high-density polyethylene geomembrane was made of high-density polyethylene resin which was extruded to extrusion sheet. The use of HDPE in landfill applications has set new standards for performance, durability, ease of installation and value. The specifications of the HDPE geomembrane are presented in Table 1.

Landfill problems

In Mavallipura landfill, simply waste has been piling up in huge pits, around 40 m deep and spread over a few hectares. Since 2007, the landfill was receiving around 1000 tonnes daily MSW from the city (approximately a total of four million tonnes have been dumped at the landfill site so far). The unprocessed waste piled high in the pits resembles a large hillock (Fig. 2). This waste was brought from Bangalore city and dumped by the municipality and contractor's trucks. This waste include newspaper, junk mail, food waste, raked leaves, dust grass clippings, broken furniture, abandoned materials, sewage sludge, industrial refuse, street sweepings, etc. The main issue arising in the landfill was the discharge of leachate which was formed by percolation of rain water through the waste and thus becoming contaminated with various biodegradable and



a Mavallipura landfill



b Landfill plan showing leachate network **c** Composting yard plan

Fig. 1 Mavallipura details

Table 1 Specifications of HDPE geomembrane in landfill site

Parameters	HDPE geomembrane
Thickness	1 mm
Density	0.940 g/cm^3
Tensile strength	$\geq$ 25 MPa
Elongation at break	$\geq$ 550%
Permeability coefficient of water vapour	$< 1.0 \times 10^{-16}$ m/s
Oxidation induction time in 200 °C, O_2, 1 atm	$\geq$ 20 min
Tangential breaking strength	$\geq$ 110 N/mm
Puncture resistance	320 N
Tear resistance	125 N
Carbon black content, range	2–3%

nonbiodegradable pollutants. The subsequent movement of the leachate into the surrounding soil, surface water or groundwater resulted in severe pollution problems.

Sampling and physicochemical analysis

Figure 3a shows a close view of sample locations marked on Google earth map. Figure 3b shows the close view of sample locations at Mavallipura landfill site. A total of 3 samples one from landfill leachate and two from nearby surface water bodies were collected.

Fig. 2 Waste covered with HDPE geomembrane sheets in landfill site

a View of sampling locations points on Google Earth

b Landfill leachate **c** Pond **d** Well

Fig. 3 The site map and the sampling locations across leachate pond, storage and groundwater sources

In order to observe the spatio-temporal variations of the geochemistry of leachates and ground waters, one undiluted representative leachate sample (L, leachate collected directly from landfill) and another two samples of water from the nearby pond (P) and open well (G) were collected from downstream of Mavallipura landfill site in the month of April 2012. After the sample collections, this landfill site was abandoned and was restricted for any further treatment and disposal due to local agitation, therefore further sampling was not possible, and the analysis was carried out only for one season samples. The pH and electrical conductivity (EC) were recorded on site at the time of sampling with digital pH meter and digital EC meter, respectively. For the analysis of biological oxygen demand (BOD), samples were collected in 300 ml capacity BOD bottles. For heavy metal analyses, samples were separately collected in pre-washed polyethylene containers of 100 ml capacity and acidified (few drops of concentrated nitric acid were added to the leachate sample) onsite to avoid precipitation of metals. The samples were then transported in cooler boxes at temperature below 5 °C and transported immediately to the laboratory. The leachate samples were stored in refrigerator at 4 °C before proceeding for the analysis and during laboratory analysis, various ionic parameters were analysed. The analysis was carried out according to standard methods for examination of water and wastewater [18]. Table 2 shows the methods of analysis used for different parameters of leachate.

Contaminated soil collection

A contaminated soil was collected from an open dump located at the Mavallipura dumping yard, Bangalore. The samples were collected for every 1-m interval (0–1, 1–2 and 2–3 m) up to the full depth of the dump of about 3 m and the sampling was done with a 150-mm diameter manual auger. Approximately 100 kg of soil sample was collected from each location. For comparison purposes, bulk sampling at selected locations was carried out using JCB excavator machines and about 100 kg was collected by quartering method. Sample temperatures were measured immediately with a thermometer. Each sample was bagged in double plastic bags and labelled. All samples were transported to the laboratory where pH (of 1:10 water extract) and moisture content (at 105 °C) were determined. Then the samples were air dried by spreading on polythene sheets. The dried samples were screened into > 20 mm, 20–2 mm and < 2 mm fractions with a mechanical vibrating screen. The first two fractions were further segregated manually into individual constituents. The soil fraction (< 2 mm) was analysed for density, Volatile Organic Matter (VOM) at 550 °C, and ash content. Total organic carbon (TOC) of these samples was determined using a solid model TOC Analyzer (Micro C, Analytic Jena. Germany). The composition of the soil is 29% clay, 19% silt and 52% sand. The soil type as per Indian Soil Classification System was observed as clayey sand (SC). The index properties of this soil are given in Table 3.

Column test

For assessing the contaminant transport parameters of metal ions through soil, the column tests were conducted according to the specifications given in the literature [4, 9,

Table 2 The methods of analysis of different parameters of leachate

Parameters	Instrument used
pH	pH meter
Conductivity, μS/cm	Conductivity meter
TDS, mg/l	TDS meter
Calcium, mg/l	EDT titration
Alkalinity, mg/l	EDT titration
Sodium, mg/l	flame photometer
Potassium, mg/l	flame photometer
Nitrate, mg/l	Spectrophotometer
Heavy metal	Absorption spectrophotometer

Table 3 Characteristics of contaminated soil

Soil	Dry density (g/cc)	Porosity (n)	Coefficient of permeability (cm/s)
Sandy soil	1.9	0.40	1.5×10^{-4}
Silty clay	1.7	0.53	1.16×10^{-5}
Hard clay	1.6	0.65	7.5×10^{-6}

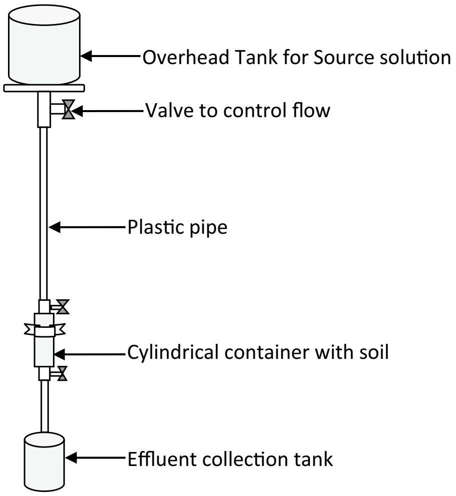


Fig. 4 Schematic diagram of Column test apparatus

19]. The constant source concentration test condition was used to conduct the column experiments. The apparatus was designed to simulate the boundary conditions of a typical landfill liner system which is shown in Fig. 4. The apparatus consists of an overhead tank mounted on a stand fixed to the wall and the soil column is connected to the overhead tank through a plastic pipe to allow seepage of solution through soil sample. Initial conditions were set as the moisture content and density of test samples as those during compaction. The constant source concentration for the influent solution was taken 100 ppm as the metal concentrations of selected chemical species were less than 100 ppm both in soil and leachate. The diameter and height of the columns were 4 cm and 10 cm, respectively. The hydraulic gradient adopted for the test was 10. The soil sample was placed in the column and the overhead tank was filled with water. The soil sample was completely saturated and then the permeability of the soil sample was measured. When steady flow conditions were established the simulated leachate was placed in the overhead tank and could pass through the soil sample. The source concentration was maintained constant throughout the test by adding same concentration of solution in the overhead tank and to maintain a constant head. The effluent samples were collected at regular time intervals till the required effluent concentration was obtained. The effluent volume was monitored at regular time intervals and concentrations were determined using Atomic Absorption Spectrophotometer (AAS).

Batch leach test

Batch leaching tests were conducted on the contaminated soil (according to ASTM D3987) to know the initial concentrations of different contaminants present in the soil. An amount of 20 g of dry soil sample was mixed with 100 ml of distilled water. The mixture was shaken by a laboratory shaker for about 24 h. The solution was then centrifuged, and supernatant solution was collected. The supernatant solution was then filtered through a 0.45-μm pore size membrane filter. The concentrations of metals presented in the effluent solution were then measured by AAS and their quantities in the soil were estimated.

Advection–dispersion–reactive equation

The mass of conservative contaminant is transported by advection and dispersion processes. The governing linear partial differential equation for one-dimensional linear advection–dispersion–reactive process in vertical direction z for the unsteady state contaminant transport in a saturated homogeneous porous medium with uniform velocity field is given as:

$$\mathrm{R}\frac{\partial \mathrm{C}}{\partial \mathrm{t}} = D_Z\,\frac{\partial^2 \mathrm{C}}{\partial \mathrm{Z}^2} - \bar{v}_z\frac{\partial \mathrm{C}}{\partial \mathrm{Z}} \tag{1}$$

where *R* is the dimensionless retardation factor defined as the ratio of the mean velocity of non-adsorbed solute to that of the retarded solute, *c* is the concentration of contaminant in the medium ($\mathrm{M/L^3}$), $\partial \mathrm{c}/\partial \mathrm{t}$ is the change in concentration of contaminant with time ($\mathrm{M/L^3/T}$), $\mathrm{D_z}$ is the longitudinal dispersion coefficient in *z* direction which takes into account both the molecular diffusion and the mechanical dispersion and is along the direction of flow and considered constant in this analysis, $\partial \mathrm{c}/\partial \mathrm{z}$ is the change in concentration of contaminant with depth i.e. concentration gradient ($\mathrm{M/L^3/L}$), and $\bar{\mathrm{v}}_\mathrm{z}$ is the average linear velocity in direction *z* which was considered steady and uniform in time and space. In Eq. (1) the term on the left side of the equation represents the change in concentration with time whereas the first term on the right-hand side accounts for the dispersion process and the second term for the advection process. In this formulation, the adsorption process induces retardation of the contaminant transport and so the advance of contaminant front is reduced as a result of sorption of the contaminant mass from solute to solid part of the porous medium.

Estimation of contaminant transport parameters

For the continuous point source of the contaminants, leaching across the soil surface boundary for infinite duration, Nelson [20] developed the analytical solution of Eq. (1) which gives the spatial and temporal distribution of concentration of contaminant. They used initial condition of $\mathrm{c\ (z, t=0)=0}$ for $\mathrm{z \geq 0}$; and the Diriclet boundary conditions of $\mathrm{c\ (z=0, t)=c_o}$ for $\mathrm{t \geq 0}$ and $\mathrm{c\ (z=\infty, t)=0}$ for $\mathrm{t \geq 0}$ and solution was obtained as:

$$\mathrm{c(z,t)} = \frac{\mathrm{C_0}}{2}\left[\mathrm{erfc}\left(\frac{\mathrm{R\,z\ -\bar{v}_z t}}{2\sqrt{\mathrm{D_z\,R\,t}}}\right) + \exp\left(\frac{\mathrm{\bar{v}_z z}}{\mathrm{D_z}}\right)\mathrm{erfc}\frac{\mathrm{R\,z+\bar{v}_z t}}{(2\sqrt{\mathrm{D_z\,R\,t}})}\right] \tag{2}$$

where $\mathrm{c_0}$ is the concentration of the contaminant at the soil surface i.e. upstream boundary. The last boundary condition can only be appreciated mathematically. However, this is required to arrive at the analytical solution (Eq. 2). The term erfc is the complementary error function which is equal to $1-\text{erf}$. For estimation of contaminant transport parameters, the method as reported by Rowe et al. [19] was used. A computer program was prepared for the above equation using MATLAB v7 software tool to perform the iterations. The experimental values of the seepage velocity ($\bar{v}_z$), thickness of the soil sample (z), time periods (t) and the effluent concentrations along with the trial values of dispersion coefficient and retardation factor (R) were incorporated into the computer program. By running this program, the theoretical effluent concentrations were determined, and the plot was generated with the theoretical breakthrough curve. The theoretical effluent concentrations were compared with experimental values and the iterations were continued till the theoretical curve matched with the experimental values.

Results and discussions

Physico-chemical parameter analysis

The important factors which influences the leachate quality such as municipal solid waste composition, elapsed time, temperature, moisture and available oxygen. Generally, the leachate quality with similar waste types may be different in different landfills located in varied climatic regions. Furthermore, operational practices in landfills also influence the leachate quality. The results of physicochemical characteristics of the leachates and water bodies from Mavallipura landfill are presented in Table 4.

The pH value of leachate sample of the landfill site was observed to be 7.4. The pH values of the pond and well water samples were found to be 8.4 and 7.5, respectively. These variations were likely caused by several factors, such as rain water infiltration and dilution effects. Additionally, the influx of contaminants from natural and anthropogenic activities like percolation of solid waste leachates and other land uses can also affect the pH values. However, these pH values were within what would be considered a relatively normal band [21].

The Mavallipura leachate sample was found to have significantly high alkalinity values. The high alkalinity observed reflects the level of biodegradation process taking place within the disposal site. The presence of significant amounts of ash and slag in Mavallipura landfill site is from the combustion of wood, agricultural residues and peat. These components are known to increase alkalinity greatly in leachates. The high concentration founds to be high in water samples this could be due to the reduced solubility of heavy metals and have effect on health. Also, this leads to unpleasant odour in the water sample and is unacceptable for many users. The high alkalinity values observed, therefore, imply that the groundwater is contaminated.

Mavallipura landfill leachate sample was found to have considerably high concentrations of all the major anions like chlorides, nitrates, sulphates as comparison with these in samples from other sites. The concentration of chloride highest, while sulphate concentration was the lowest. The high chloride content in the leachate sample reflects the significant presence of soluble salts in the municipal solid waste materials of the study area. The high chloride content in Mavallipura landfill leachate sample is attributed to the large amount of sewage, agricultural and another animal waste deposited in the site. Prior to anaerobic activity sulphate is converted to sulphide and metal sulphide precipitates in leachate sample.

Table 4 Physicochemical characteristics of the leachate

Characteristics	L	P	G
pH	7.4	8.4	7.5
Conductivity, µS/cm	4120	2500	1362
Total dissolved solid (TDS), mg/l	2027	1447	703
Chemical oxygen demand (COD), mg/l	10,400	1080	440
Biochemical oxygen demand (BOD_5), mg/l	1500	105	3
Sulphate, mg/l	40	10	7
Chloride, mg/l	660	250	230
Calcium, mg/l	400	0	320
Alkalinity, mg/l	11,200	2000	300
Iron, mg/l	11.16	0.16	0.62
Copper, mg/l	0.151	BDL	BDL
Silver, mg/l	0.035	0.026	0.051
Cadmium, mg/l	0.035	BDL	BDL
Chromium, mg/l	0.021	BDL	BDL
Lead, mg/l	0.3	BDL	BDL
Zinc, mg/l	3	1	0.4
Nickel, mg/l	1.339	BDL	BDL
Sodium, mg/l	3710	1676	88
Potassium, mg/l	1675	1078	46
Nitrate, mg/l	22.36	0.18	1.09
Ammonia nitrogen, mg/l	1803	0.5	0.5

Parameter: Normal range for drinking water

Hardness 5–50 ppm; pH 6.5–8.2; Copper < 0.5 ppm; Iron < 0.2 ppm

Phosphate < 0.03 ppm; Chlorine < 0.5 ppm; Ammonia < 1 ppm; Chromium < 0.5 ppm

Sulphates were found to be high in the leachate sample due to the decomposition of organic matter, soluble waste, such as construction wastes or ash, synthetic detergents and inert waste, such dredged river sediments. Similarly, nitrates represent the most oxidized form of nitrogen found in natural system. It is often regarded as an unambiguous indicator of domestic and agricultural pollution. In leachate sample it was formed primarily as a result of oxidation of ammonium to nitrite and subsequently, to nitrates by nitrification process.

The constituent's calcium, magnesium, sodium and potassium are considered generally to be major cations typically present in leachate. In leachates, these are sourced usually from the degradation of organic materials and the dissolution of inorganic wastes such as concrete, plaster and tiles. Sodium and potassium are both present at considerably high concentrations in the entire samples of this investigation. The sodium and potassium are not affected significant by microbiological activities within the landfill site.

Calcium is one of the most common cations found in ground water aquifers, as it dissolves from rocks, such as limestone, marble, calcite, dolomite, gypsum, fluorite and apatite. In addition, with magnesium, it is one of the principal cations associated with water hardness. Calcium concentrations were noticeably high in well sample.

Concentrations of heavy metals were found to be low in leachates samples except for Fe and Zn. Heavy metals appear in the landfill from sources like batteries, consumer electronics, ceramics, light bulbs and glass can all introduce metal contaminants into the solid waste stream. Concentration of heavy metals in a landfill is generally higher at earlier stages because of higher metal solubility as a result of low pH caused by production of organic acids. It is now recognized that most trace elements are readily fixed and accumulate in soil sand because this process is largely irreversible, repeated applications of amounts in excess of plant needs eventually contaminate the soil and may either render it non-productive or the product unusable. Although plants do take up the trace elements, but the uptake is normally so small that this alone cannot be expected to reduce appreciably the trace element.

The quality assurance mainly depends on the accuracy and reliability of the information used. This is very necessary for water quality assessments in which the compositional characteristics of a sample can change rapidly when influenced by environmental factors. Hence, it is important to process the sampling and analyse to achieve high-quality analytical results. It is well documented that within water bodies samples, the number of positively cations should balance the number of negatively charged anions. Cation–anion balance is calculated by comparing the total charge of the positive-charged ions (cations) with the total charge of the negative-charged ions (anions). Ionic balance error for ground water samples should not exceed 5% unless where the TDS value is less than 5 mg/l, in which case a higher error is tolerable (UNEP/WHO, [22]). The ion compositions were plotted for all water bodies samples with an error of ion balance within the ± 5% range. Major cations such as Na^+, K^+, Ca^{2+} and Mg^{2+} and major anions such as HCO^{3-}, CO_3^{2-}, SO_4^{2-} and Cl^- were plotted in a hydrochemical trilinear diagram, also known as a Piper diagram. Figure 5 shows the Piper diagram, composition of leachate samples (L) according to their major cations and anions. The ionic balances were calculated and examined for each sample. The major anions considered were total alkalinity, chlorides, sulphates and nitrates, while the major cations included are calcium, magnesium, sodium and potassium. Leachate sample whose error of ions was within ± 5% were plotted.

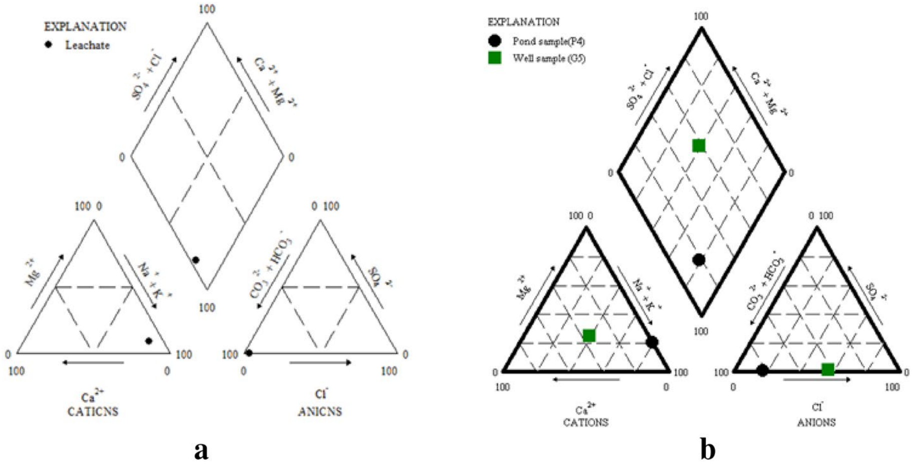


Fig. 5 Piper diagram, composition of **a** leachate samples; **b** water bodies samples

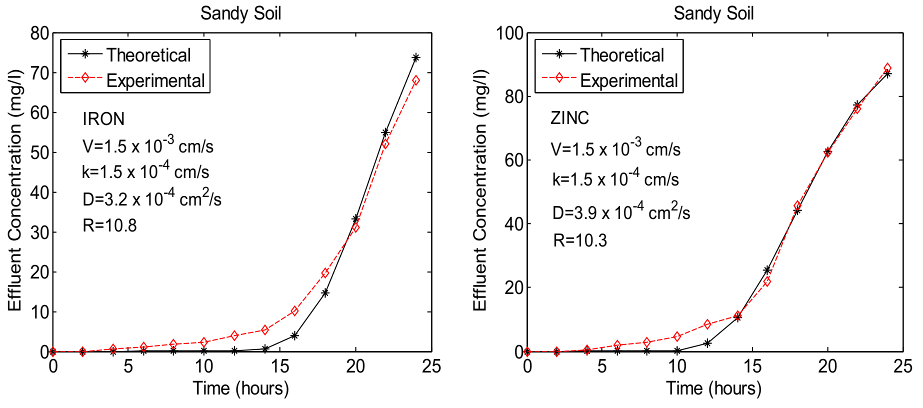
The piper diagram reveals composition of different ions in percentage. A major conclusion can be drawn from the piper diagram of the collected leachate samples. More importantly it indicates predominance of select cations as Na^+ and K^+ in comparison to calcium and magnesium. Bicarbonates and carbonates are the dominant anion found in the leachate samples compared to sulphates and chlorides. High chloride builds up are possibly due to leaching of percolating water from the MSW. The analyzed leachate sample can be thus categorized as the $Na–HCO_3$.

Also, piper diagram identifies the hydrogeochemical facies. Based on piper diagram a few conclusions can be inferred of the collected water's bodies samples. Firstly, it indicates large percentages of the samples within the $Ca–SO_4$ category followed by $Na–HCO_3$ type. Secondly, bicarbonates and carbonates dominated the anions compared to chloride. However, anions like sulphates were very meager in concentrations in relation to other anions. Chloride content is high in ground water can be possibly domestic effluents, fertilizers, septic tanks, and leachate. Moreover, the above plot shows Ca^{2+} concentrations being noticeably high in well sample. However, Na^+ and K^+ dominated the pond samples.

Contamination transport model

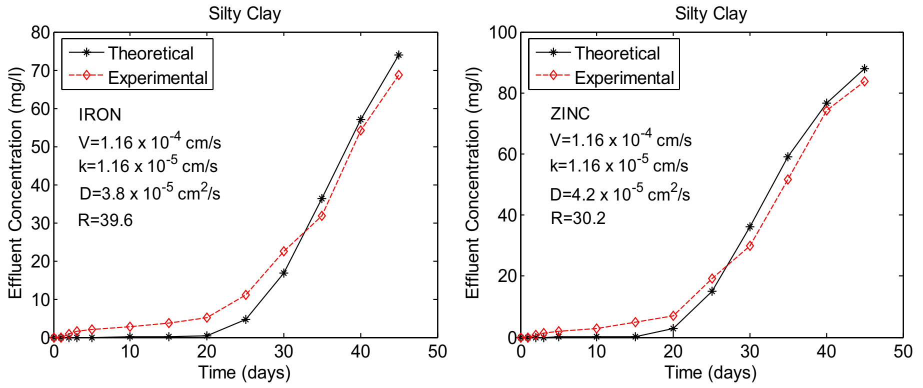
Once the landfill leachate is released into the subsurface, contaminants will interact physically, hydrologically and biochemically with both the water bodies and the soil matrix. A complete transport model must therefore account for multispecies physical and biochemical transport processes. *Fluidyn* groundwater model-POLLUSOL is the most powerful software package designed specifically for continuous simulating 3-dimensional groundwater flows and predicting pollutant dispersion from a source through porous medium. Flow equations for flow into porous medium are employed using conservation of total fluid mass principle and Darcy's law. Solute transport equation is derived from conservation of mass for a single solute or multiple solute species that may decay or adsorb in the porous medium. Simulation was carried out simultaneously for groundwater flow and pollutant dispersion using the built-in solver of *fluidyn*-POLLUSOL which is based on the finite volume method. The simulations were carried out in steady state mode. The generalized Navier–Stokes equations, for porous media, were solved to obtain the groundwater flow field. The contaminant transport parameters of the species of interest in the porous media were determined through column tests using the procedure as described in "Contaminated soil collection" and "Column test" sections. These parameters are given as input data for generating output from the *fluidyn*-POLLUSOL model. The contaminant transport parameters determined by matching the theoretical breakthrough curve with the experimental results are shown in Figs. 6, 7 and 8. The contaminant transport parameters of the test samples along with their physical properties are as given in the Table 5. These parameters are given as input data for generating output from the fluidyn-POLLUSOL model.

It can be observed that the dispersion coefficients of hard clay are much lower and distribution coefficients are much higher than those of silty clay and sandy soil even though the density of hard clay sample is less than the other samples. This effect is due to the presence of more clay fraction presented in the hard clay which causes less hydraulic conductivity and more adsorption capacity. From the test results (Figs. 6, 7, 8), it is



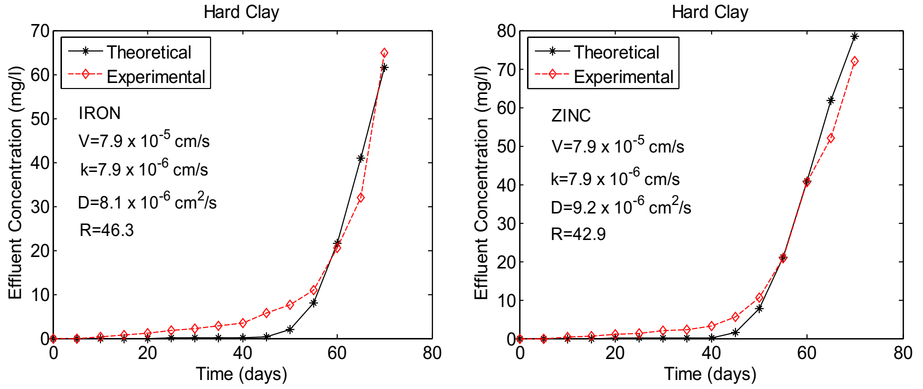
a Iron ion migration through sandy soil **b** Zinc ion migration through sandy soil

Fig. 6 Migration of ions through sandy soil



a Iron ion migration through silty clay **b** Zinc ion migration through silty clay

Fig. 7 Migration of ions through silty clay



a Iron ion migration through hard clay **b** Zinc ion migration through hard clay

Fig. 8 Migration of ions through hard clay

Table 5 Contaminant transport parameters of column test samples

Soil	Metal ion	Dispersion coefficient (cm^2/s)	Retardation factor (R)
Sandy soil	Iron	3.2×10^{-4}	10.8
	Zinc	3.9×10^{-4}	10.3
Silty clay	Iron	3.8×10^{-5}	39.6
	Zinc	4.2×10^{-5}	30.2
Hard clay	Iron	8.1×10^{-6}	46.3
	Zinc	9.2×10^{-6}	42.9

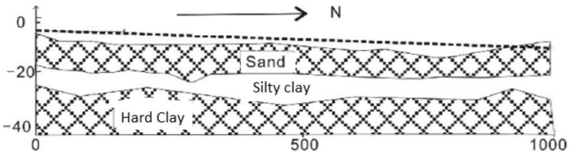


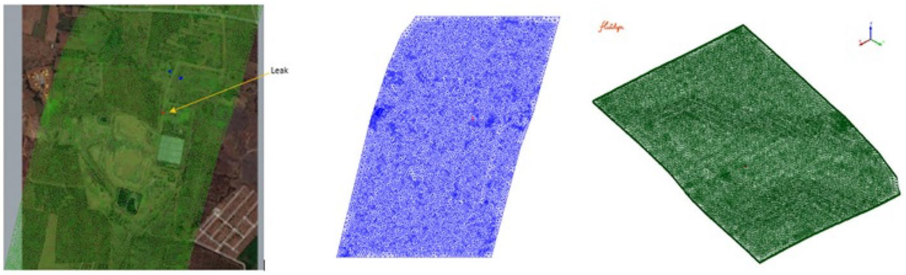
Fig. 9 Conceptual cross-section of the modeled area

evident that the dispersion coefficients of Iron are less than that of Zinc whereas the retardation factors of Iron are higher than Zinc in all the three soils studied. This may be due to the less mobility of Iron caused by higher specific gravity of Iron compared to Zinc. The specific gravity of Iron varies from 7.6 to 7.9 whereas it varies from 6.9 to 7.2 for Zinc. This affinity order of Cu and Zn ($Cu > Zn$) for soils studied is similar to that obtained in the previous studies of Matos et al. [23], Young et al. [24], Soares [25], Vega [26], Nascentes et al. [12] and Korf et al. [9]. However, the differences in the transport parameters of iron and Zinc are relatively very less in the soils studied.

Fluidyn's groundwater model

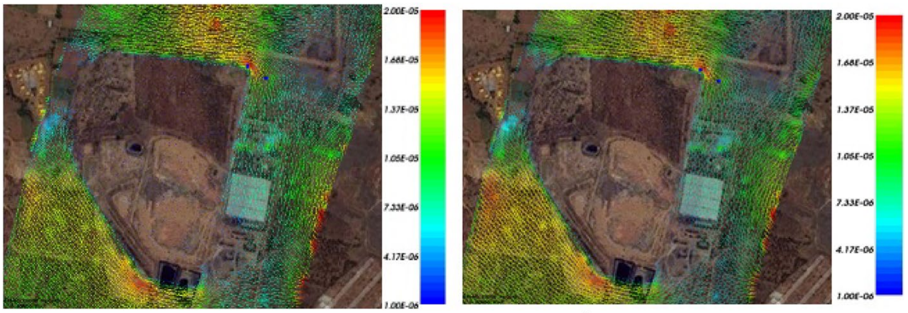
The contour data of the terrain (topo map) of Mavallipura landfill, is used to generate the computational domain. The borehole data, at the landfill site, was used for modelling the entire domain. According to the borehole data there are three layers of soil at the location, namely sandy, silty clay and hard clay occurring in the same order. Sandy soil was at the top and hard clay was at the bottom next to the impervious bed rock as shown in Fig. 9.

The building cover is considered by making the surface, on which the building stands, impervious to the rainfall. The computational domain covers an area of 0.372 sq. km and a soil depth of 6 m. The geometrical model of the terrain and the leak detected are shown in Fig. 10a. The leachate can seep into the soil at a location next to the landfill. Initially, the location of the leak is selected by assuming. The area of the leak is 31.5 m^2 (approx.). The volume flow rate of the leak is arbitrarily chosen to be 10 ml/s. The selection of the pollutants to be modeled was based on the corresponding concentrations in site specific leachate samples, susceptibility to natural attenuation and drinking water standards. The physicochemical analysis revealed that iron (Fe) and zinc (Zn) would be the most critical indicators. So, concentration of the pollutants being considered (Fe and Zn), in the leachate at the leak, was set to 1. Also, the simulation was done for a worst-case scenario.



a Computational domain **b** Unstructured mesh at the top surface & 3D view of the mesh at top surface

Fig. 10 Finite volume mesh



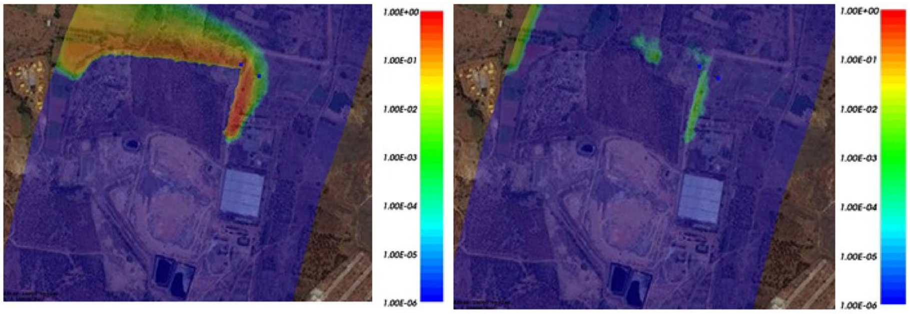
a Average rain fall conditions **b** Monsoon conditions

Fig. 11 Groundwater velocity vector plots in the sandy soil layer

Rainfall data was collected from the Indian Meteorological Department. The average annual rainfall in Bangalore is 978 mm. This data has been used to provide constant and uniform rainfall over the computational domain, for the first simulation. The average monthly rainfall during monsoon is ~ 171 mm. This data has been used to provide constant and uniform rainfall over the computational domain, for the second simulation. A nearly uniform unstructured mesh was generated for the study. The mesh created is made of layered unstructured cells (prismatic). The mesh used contains 333,536 cells (Fig. 10b).

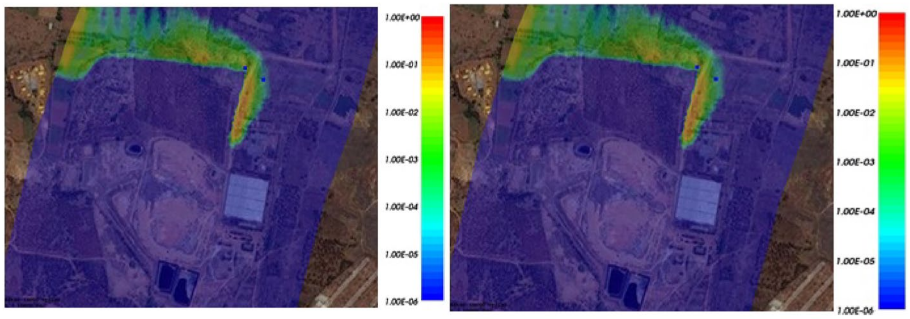
The groundwater velocity vector plots in the sandy soil layer, for average rainfall and monsoon conditions, are shown in the Fig. 11. The velocity vectors in the top half of the silty clay layer is like the ones in the sand layer but lesser in magnitude by about one order. The velocity vectors in the lower half of the silty clay layer and the hard clay layer are almost nonexistent. The blue dots in the Figs. 12, 13, 14, 15, 16, 17, 18 and 19 indicate the locations of the water bodies.

As shown in Fig. 9, silty clay is the thin layer. The thin horizontal layer of silty clay extends across the flow domain (Figs. 13, 14, 15), it causes the contaminants to move through the flow system almost entirely in this thin layer. The total travel time would be one-fifth and sandy layer of higher conductivity has a 100-fold larger value of k than the rest of the system and exerts a very strong influences on the migration patterns and velocity distribution.



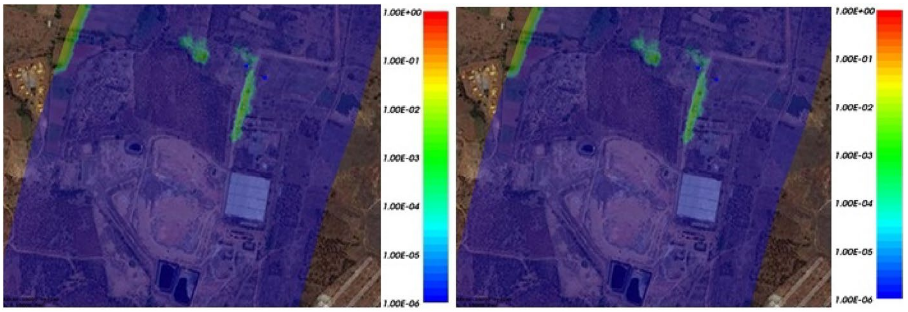
a Under average rainfall conditions **b** Under monsoon conditions

Fig. 12 Normalized concentration of zinc in the sandy layer



a Under average rainfall conditions **b** Under monsoon conditions

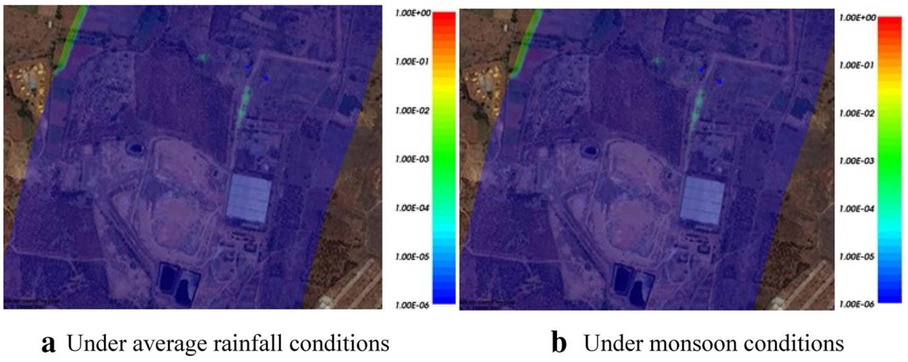
Fig. 13 Normalized concentration of zinc in the top half of silty clay layer



a Under average rainfall conditions **b** Under monsoon conditions

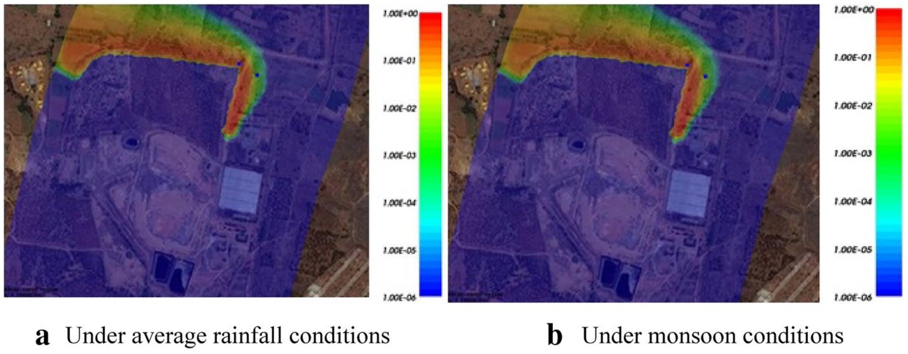
Fig. 14 Normalized concentration of zinc in the bottom half of silty clay layer

The velocity vectors clearly show how the pollutants from the leak reach the water bodies. While for the pond (the blue dot towards north-west), the pollutant reaches merely by means of advection, for the well (the blue dot towards southeast), its pollution would depend also on the diffusion. This result justifies positioning the leak just after the building cover such that crosswind diffusion is free to take place and achieve maximum dispersion. The distribution of zinc and iron in the various layers, under average rainfall and monsoon conditions, is shown in the Figs. 12, 13, 14, 15, 16, 17,



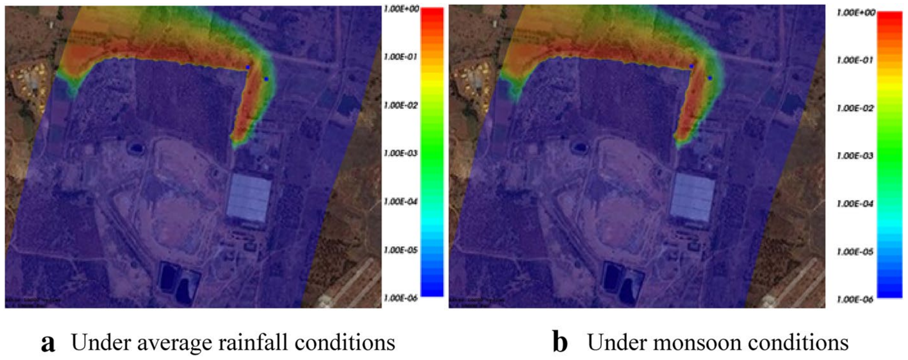
a Under average rainfall conditions **b** Under monsoon conditions

Fig. 15 Normalized concentration of zinc in the hard clay layer



a Under average rainfall conditions **b** Under monsoon conditions

Fig. 16 Normalized concentration of iron in the sandy layer



a Under average rainfall conditions **b** Under monsoon conditions

Fig. 17 Normalized concentration of iron in the top half of the silty clay layer

18 and 19. The plots for average rainfall and monsoon conditions do not show much difference. However, the rainfall may have a linear effect only.

Several forces react with the leachate migration resulting in changes in chemistry and reduction of strength from the original release. The physical forces like filtration, sorption, advection and dispersion and chemical forces such as oxidation–reduction, precipitation–dissolution, adsorption–desorption, hydrolysis and ion exchange, and biological reaction like microbial degradation. These reactions will depend on the soil stratification of the landfill, the hydraulic characteristics of the ground water system, and the chemistry of the

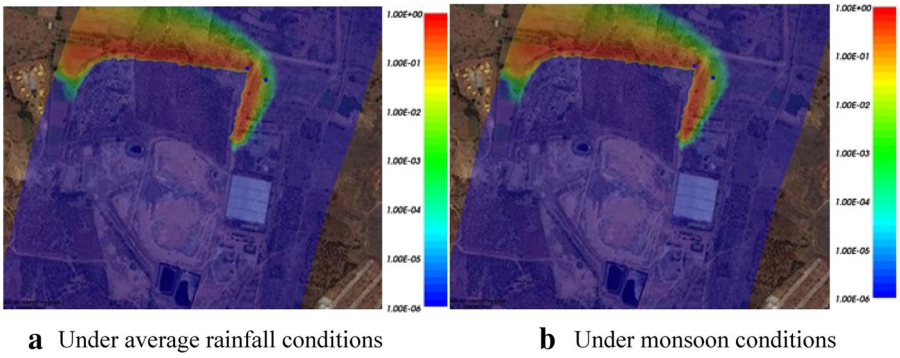


Fig. 18 Normalized concentration of iron in the bottom half of the silty clay layer

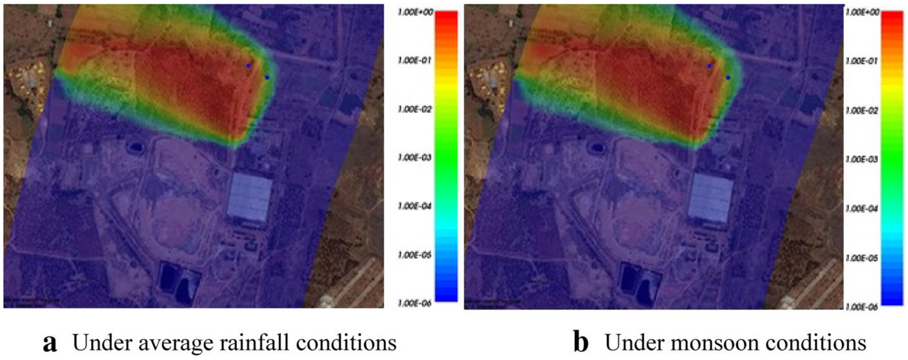


Fig. 19 Normalized concentration of iron in the hard clay layer

leachate. Also, these reactions have the capability to reduce the potential impact to groundwater and surface water.

Mavallipura landfills are constructed with liners that contain leachate, and leachate collection systems that collect it. But due to unsegregated and mismanagement of landfill, leads to puncture the linear system in the landfill. This resulted in creating significant leachate-contaminated to the pond water and openwell water environment, it will migrate downward through the unsaturated zone until it reaches the saturated zone. Finally, leachate then will follow the hydraulic gradient of the groundwater system.

Conclusions

Results of physicochemical parameters and determined the contents of heavy metals leaching from Mavallipura landfill suggest an exceptional pollution. High contents of organic pollutants and heavy metals found on landfill sites, as well as the concentration of these metals in ponds and open wells in Mavallipura. Iron and zinc concentrations are higher in leachate sample.

Physicochemical analysis reveals that there is high contamination of organic & inorganic constituents. Heavy metals concentration was in trace only indicating that the waste dumped is predominantly municipal waste. Based on physicochemical analysis the quality of water was found to be unsuitable for drinking purpose.

The study revealed that the concentration of MSW materials in the landfill site had systematically polluted the soil and water bodies (surface water and groundwater) over time. The effect of such pollution as assessed from the study declined away from the polluting source. This implied that the surface and groundwater bodies were more dependent on proximity to landfill sites. This dependence may be due to the influence of topography, type, state of waste disposal system and to some extent, the hydrogeology of the site.

Based on the groundwater model, the results revealed that the zinc concentration observed was being limited to upper layers of the soil, owing to the large time required in seeping through the layers and its radioactively decaying tendency. At the same time, the iron concentration showed the maximum dispersion that could be achieved by any non-decaying pollutant. The results make it evident that under the assumed conditions, it is highly probable that the groundwater flow could cause contamination of the water bodies in the vicinity of the landfill due to a leak at the landfill. This leak area has to be repaired either by laying a new liner or by using some cement grouts to close the leak. Constructing a new liner for the leak area may be difficult as it is required to excavate the landfill and hence grouting is a better option to close the leak area with cementing grouts. As the water bodies surrounding the landfill are already contaminated, it is recommended to treat them using pump and treat technique.

Abbreviations

R: dimensionless retardation factor; c: concentration of contaminant; $\partial c/\partial t$: change in concentration of contaminant; $\partial c/\partial z$: change in concentration of contaminant; $\bar{v}_z$: average linear velocity; t: time period; D_z: dispersion coefficient.

Authors' contribution

BPN entire manuscript written and detailed analysis and modelling is done. JS contribution as mentioned below: column test, batch test. RKM contribution as mentioned below: mathematical equation developed. All authors read and approved the final manuscript.

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Competing interests

The authors declare that they have no competing interests.

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